Nitrate and Chloride Loading to Groundwater from an Irrigated North-Central U.S. Sand-Plain Vegetable Field

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ABSTRACT

Groundwater pollution and associated effects on drinking water have increased with the expansion of irrigated agriculture in north-central U.S. sand plains. Controlling this pollution requires an ability to measure and predict pollutant loading by specific agricultural systems. We measured NO_3 and Cl loading to groundwater beneath a Wisconsin central sand plain irrigated vegetable field using both a budget method and a new monitoring-based method. By relying on frequent monitoring of shallow groundwater, the new method overcomes some limitations of other methods. Monitoring-based and budget methods agreed well, and indicated that loading to groundwater was 165 kg ha^{-1} NO_3–N and 111 kg ha^{-1} Cl for sweet corn (*Zea mays* L.) in 1992, and 228 kg ha^{-1} NO_3–N and 366 kg ha^{-1} Cl for potato (*Solanum tuberosum* L.) in 1993. Nitrate N loading was 56 to 60% of available N, or 66 to 70% of fertilizer N. Sweet corn NO_3 loading was about typical for this region, but potato NO_3 loading was probably 50% greater than typical because heavy rains provoked extra fertilizer application. Our results imply that typical NO_3–N loading would be 119 kg ha^{-1} for sweet corn and 203 kg ha^{-1} for potato, even with strict adherence to University Extension fertilizer recommendations. To keep average groundwater NO_3–N within the 10 mg L^{-1} U.S. drinking water standard, each irrigated vegetable field would need to be offset by five to eight times as much land supplying NO_3-free groundwater recharge.

Irrigated agriculture has expanded eightfold in the sand plains of the humid, north-central USA since 1970 (Bajwa et al., 1992; Mossbarger and Yost, 1989), but not without environmental cost. In particular, substantial groundwater pollution has resulted from the coupling of agricultural systems demanding large inputs of fertilizer, pesticide, and irrigation water with a physical setting that includes coarse soils and shallow groundwater. For instance, in the agricultural statistics district that contains much of the Wisconsin central sand plain (Fig. 1), 22% of domestic wells exceed the NO_3 maximum contaminant level (MCL; a U.S. drinking water standard), 23% contain detectable atrazine (2-chloro-4-ethylamino-6-isopropylamino-s-triazine), and 23% contain alachlor (2-chloro-2′-6′-diethyl-*N*-[methoxymethyl]-acetanilide) (LeMasters and Baldock, 1995). In parts of the Wisconsin central sand plain where irrigated agriculture is the dominant land use, more than 60% of wells may exceed the NO_3 MCL (Central Wisconsin Groundwater Center database, unpublished data, 2000).

A connection between agriculture and groundwater pollution is well established (Holden et al., 1992; Hamilton and Helsel, 1995). What is needed is the ability to measure and predict the mass loading of contaminants to groundwater from particular agricultural systems. Such an understanding would be useful for predicting groundwater quality, determining how much agricultural land use a given physical setting can support, and meeting groundwater quality goals.

Our objective was to determine NO_3 loading to groundwater under an irrigated vegetable field. We estimated NO_3 loading from the 1992 sweet corn and 1993 potato crops using a novel *water-year* method based on monitoring of shallow groundwater under the study field. For comparison, NO_3 loading was also estimated with a budget approach, and the same methods were applied to Cl loading.

Loading-Rate Estimation

Solute loading to groundwater in agricultural systems has often been estimated by collection and analysis of soil pore water, monitoring of drain-tile effluent, pan or monolith lysimeter studies, and mass-balance computations. Each has its own limitations.

Determining solute loading from soil-pore water extracted in situ by suction (e.g., Perillo et al., 1993) or ex situ from soil cores (Hill, 1986) entails several characteristic problems (Hansen and Harris, 1975; Haines et al., 1982; Kinniburgh and Miles, 1983; Kung, 1990a,b; Wehtje et al., 1984; Barbee and Brown, 1986). Each sample represents only a small volume of soil, so many samples are needed to characterize a heterogeneous field (Hart and Lowery, 1997). Preferential flow paths are likely to be missed by randomly located suction samplers or soil cores (Shuford and Baker, 1977; Kung, 1990a), leading to an underestimate of peak solute concentrations (Kung and Donohue, 1991). In addition, sample composition may be altered by the extraction process (Kinniburgh and Miles, 1983) or by chemical reactions in samples waiting to be collected (Nagpal, 1982). Finally, extracted pore water only provides concentration data; water flux must be determined separately to allow solute loading to be calculated.

Tile drains (e.g., Milburn et al., 1990; Madramootoo et al., 1995; Randall and Irigavarapu, 1995) can collect water from a large area, avoiding some problems of field heterogeneity and preferential flow (Hallberg et al., 1986). However, though tile effluent is often assumed to be equivalent to groundwater recharge, leachate can bypass the drainage lines (Kladivko et al., 1991), and in some cases the drain-tile effluent includes upwelling groundwater and not just field leachate (Bergström, 1987). Thus, both flow volumes and solute con-

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Abbreviations: MCL, maximum contaminant level; MLP, multilevel piezometer.

centrations can be in error. In any event, measuring loading using tile drains is only practicable in high-water-table settings.

Pan lysimeters (Tyler and Thomas, 1977; Jemison and Fox, 1994) sample from a somewhat larger area than suction samplers or soil cores, and thus are more likely to intercept preferential flow. However, they are small, and many pan lysimeters would be needed to adequately represent field heterogeneity. Installation can create substantial soil disturbance, which may introduce error, at least temporarily (Watts et al., 1991). Pan lysimeters provide both concentration and water-flux data, but often the collection efficiency is assumed to be 100%, which is not necessarily valid. Jemison and Fox (1994) determined a pan efficiency of 52% by a Br balance method.

Monolith lysimeters (Saffigna and Keeney, 1977; Saffigna et al., 1977) allow better determination of water fluxes than other methods, but edge effects on plant growth and hence transpiration can be substantial (Bergström, 1990), and the lysimeter wall can create preferential flow paths (Cameron et al., 1979). Small-diameter lysimeters force flow into vertical paths, whereas undisturbed flow paths can have a major horizontal component, for example under funnel flow conditions (Kung, 1990b). Many monolith lysimeters might be required to represent field-scale variability.

Mass-balance budget methods have been applied many times to a wide variety of crops, and the principles are well known (Meisinger and Randall, 1991). The main problem is the difficulty of estimating some important fluxes. Mineralization of soil organic N and, to a lesser degree, atmospheric inputs and outputs (Broadbent, 1981) are particularly troublesome.

Estimating solute loading to groundwater by directly measuring concentrations in groundwater, as in this study, has the potential to reduce or avoid problems due to soil water being unrepresentative of groundwater, preferential flow, soil disturbance during equipment installation, soil sample volumes that are too small to represent field heterogeneity, and water flux estimation.

STUDY AREA

The Wisconsin Central Sand Plain

The study was done in the Wisconsin central sand plain, a 6400-km^2 area characterized by level topography and a mantle of coarse-grained Pleistocene glaciofluvial and lacustrine sediment frequently more than 30 m thick. Upland soils are extremely well-drained, averaging 93% sand and 1% organic matter in Ap horizons, and 98% sand and 0.1% organic matter in subsoils. Pleistocene sediment overlies crystalline rock, and in places, sandstone. Only the Pleistocene sediment is significant as an aquifer because the other units have limited permeability or extent. Aquifer materials are sandy and consist of 92 to 95% quartz; other minerals include feldspar, mica, and mafic minerals. The organic matter content is typically less than 0.1%. Groundwater in the area is well oxygenated and NO_3^- is conserved, except for the deepest part of the aquifer at a few locations (Kraft et al., 1999; Stites and Kraft, 2000).

Annual precipitation averages 795 mm, with a 10% probability of less than 625 or more than 1020 mm. Approximately

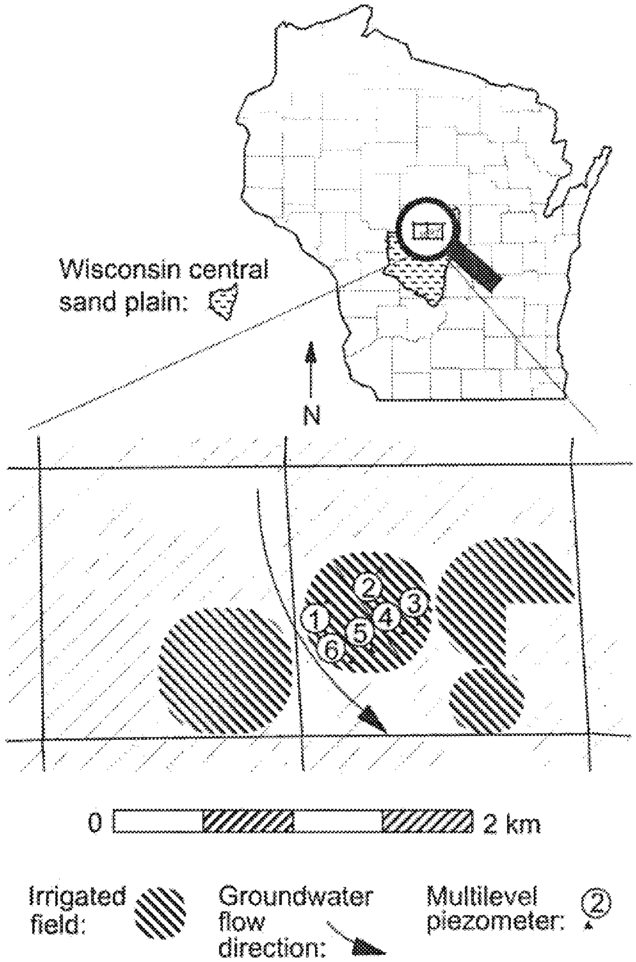

Fig. 1. Location map and distribution of multilevel piezometers in the study field.

60% of the annual precipitation falls in May through September. Weeks et al. (1965) and Holt (1965) estimate that, of 760 to 790 mm of precipitation, 510 to 560 mm goes to evapotranspiration, 230 to 255 mm to groundwater recharge, and 25 mm to direct runoff. The average frost-free growing season is 133 d (Bartelme, 1977).

The irrigated vegetable rotation in the area consists of 1 yr of potato and 2 to 3 yr of snap bean (*Phaseolus vulgaris* L.), sweet corn, field corn (*Zea mays* L.), soybean [*Glycine max* (L.) Merr.], or pea (*Pisum sativum* L.), in that order of frequency (D. Sexson, personal communication, 1999). Typical fertilizer N applications to these crops are (respectively) 258, 110, 200, 180, 67, and 67 kg ha^{-1}. Most Cl arrives in a single spring application of KCl. Manure is rarely applied. Other inputs to this system are presented in Stites and Kraft (2000). Field operations frequently begin as early as March, when light snow cover may still be present, with the broadcast of KCl fertilizer. Planting usually occurs between mid-April (for early potato) and the beginning of June. Fertilizer materials are applied at planting time and two to four additional times by banding and with irrigation water. Irrigation wells in or next to fields tap the shallow Pleistocene aquifer, and the water is distributed by sprinkler. Depending on weather, irrigation water is applied about 15 times per year, about 15 mm per irrigation. Irrigation increases the transpiration of fields by about 100 mm yr^{-1}. Irrigation also increases percolation and solute leaching (Weeks et al., 1965; Weeks and Stangland, 1971). Nonadsorbing solutes usually leach to the water table within a few months after application (e.g., Harkin et al., 1986; Kung, 1990a).

Study Field and Management Practices

The study field, Field 2 of Stites and Kraft (2000) (Fig. 1), is 44 ha in the north part of the Wisconsin central sand plain. Pleistocene deposits in this field consist of about 20 m of well-sorted medium sand with a 1-m interbedded fine-textured layer about 7.5 m below the surface. Groundwater occurs at 2.5 to 3 m depth, and flows southeast at about 0.1 m d^{-1}. The soil is Plainfield loamy sand (mixed, mesic Typic Udipsamment). Nitrate N concentrations in shallow groundwater under the study field and three nearby fields averaged 21 mg L^{-1} compared with 1 mg L^{-1} upgradient, and Ca, Cl, K, and Mg concentrations were 5 to 26 times greater under fields than upgradient. Residues of five pesticides were detected under the fields with a median summed concentration of 1.5 μg L^{-1} (Stites and Kraft, 2000).

The 1992 sweet corn crop received 250 kg ha^{-1} fertilizer N and 99 kg ha^{-1} Cl from KCl. Fertilizer N was about 25% more than average for area sweet corn (J. Exo, personal communication, 1993) and 49% more than the University Extension recommendation. The growing season was shorter, cooler, and slightly drier than average. The harvest was 25 Mg ha^{-1} (fresh weight, ears with husks). The 1993 potato crop received 297 kg ha^{-1} N in the north half of the field and 357 kg ha^{-1} in the south half. Both halves received 353 kg ha^{-1} Cl from KCl. The harvest was 46 Mg ha^{-1}, fresh weight. Fertilizer N was much greater than the local average and the University Extension recommendation for potato (258 kg ha^{-1}; Binning et al., 2000). The cooperating grower applied extra fertilizer N after record May and June rainfall (347 mm compared with 218 mm average) because of perceived leaching losses.

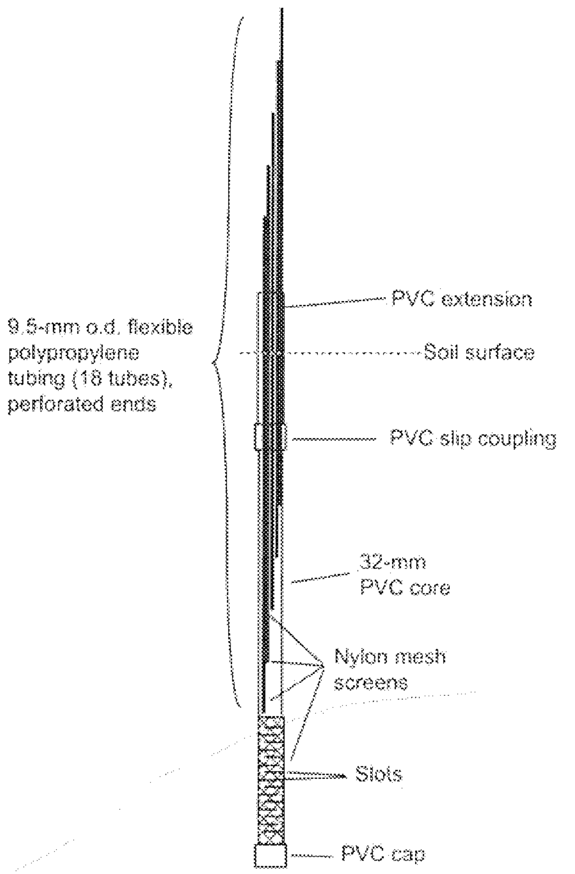

Fig. 2. Schematic view of a multilevel piezometer. Sampling points are the 18 flexible tubes and the rigid "backbone," which forms the deepest point.

METHODS

Instrumentation, Sampling, and Analysis

The study field was instrumented in 1991 with six multilevel piezometers (MLPs), each containing 19 ports over a total continuous screened length of 340 cm (Fig. 2). In early 1994, up to five additional ports were added above the existing MLPs in response to a rise in the water table after the heavy 1993 rainfall; however, these were only available for the last sampling round of this study. More detail about instrumentation and sampling is in Stites and Kraft (2000). All MLPs were placed more than one year's groundwater travel distance away from upgradient field edges (as calculated by Darcy's Law), so that a complete year of recharge from inside the field would be available within the sampled zone.

Groundwater was sampled 23 times from January 1992 through April 1994 at 2-wk to 2-mo intervals, depending on the season and on field operations. Usually only alternate MLP ports were sampled. Nitrate + nitrite N and Cl were analyzed by automated colorimetry (Lachat Instruments, 1986, 1989) in a certified laboratory. (Nitrite has been undetectable or minuscule relative to NO_3 in this setting [R. Stephens, personal communication, 1998], so we refer to NO_3–N + NO_2–N simply as NO_3–N.) Chloride and NO_3 concentration–depth profiles were constructed for each MLP on every sampling date. These were displayed sequentially, as in Fig. 3, to apply the water-year loading method.

Water-Year Loading Method

The theory of measuring solute loading by the water-year method is illustrated by Fig. 4. All groundwater recharge originating during one year in a region having area A will occupy a volume V in the aquifer. The areas of the top and bottom surfaces of V are also A. The volume of water in V is $\theta A(z_2 - z_1)$, where θ is the volumetric water content of the aquifer, and z_1 and z_2 are the elevations of the bottom and top of V. This volume of water will also contain one year's contribution of any conservative solute we are interested in. If the solute's concentration C were uniform with depth, the mass m of solute in V would be $\theta AC(z_2 - z_1)$; rearranging and accounting for the nonuniform concentration by integrating in the vertical direction yields annual loading per unit area:

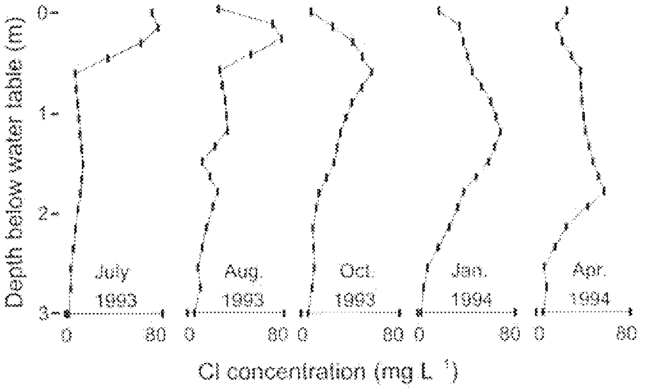

Fig. 3. Arrival and downward movement of annual Cl pulse in one study field multilevel piezometer (MLP). Five snapshots are shown of 23 profiles constructed over the period. (See text.)

$$m/A = \theta\int_{z_1}^{z_2} C\,dz. \qquad [1]$$

The challenge to implementing the water-year method is to determine z_1 and z_2. We accomplished that by using Cl from fertilizer as a fortuitous tracer. Chloride was applied to the study field as KCl fertilizer in one large dose each year, 36 to 76 d before planting. Most of the Cl was leached beyond the reach of plant roots by spring rains, as indicated by Harkin et al. (1986) and Kung (1990a) and by solute transport simulations (unpublished data, 1996). This Cl formed a pulse in the vadose zone that, in time, moved to the water table. The structure of the Cl pulse was retained in concentration–depth profiles in the saturated zone. This is illustrated in Fig. 3 for the 1993 water year. In the July 1993 profile, the leading edge of the 1993 Cl pulse is present near the water table. Subsequent profiles show the continued breakthrough and downward movement of the pulse. In the April 1994 profile, the entire 1993 Cl pulse is in the saturated zone, and a new pulse is arriving at the water table from the 1994 KCl application. In this profile, Cl minima at 0.2 and 2.6 m depth bracket the 1993 Cl pulse. These depths correspond to z_1 and z_2.

We refer to the depth range in the aquifer containing one year's recharge as a *water-year interval*, operationally defined as the depth range between Cl minima. Delineating the water-year interval in groundwater using Cl permitted monitoring data to be used to estimate loading of any solute during the corresponding year. That is, once the water-year interval was defined, calculating the mass of solute in that interval provided a measure of solute loaded during the corresponding year.

We identified water-year intervals for each MLP by comparing a time series of Cl concentration–depth profiles. When the water-year interval was within the sampled zone of an MLP, we calculated the solute load for that water year, using a discrete version of Eq. [1]:

$$m/A = \theta\sum_{p=i}^{j} C_p \Delta z_p \qquad [2]$$

where i and j are the bottom and top MLP ports in the range from z_1 to z_2, C_p is the concentration in port p, and Δz_p is the length of port p. (For ports at the boundary between water-year intervals, half of the port's length was assigned to each year.) Aquifer porosity (equivalent, in the saturated zone, to volumetric water content) was 0.40, as determined from measurements on intact cores extracted near the study field (Kraft et al., 1995).

Usually, a given water-year interval remained in the monitored zone for several sampling dates as it moved downward.

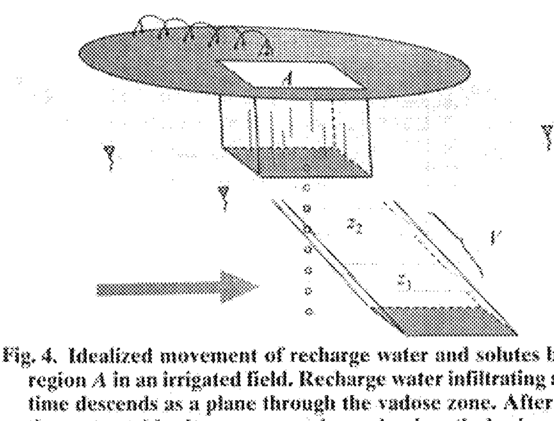

Fig. 4. Idealized movement of recharge water and solutes beneath a region A in an irrigated field. Recharge water infiltrating at a given time descends as a plane through the vadose zone. After reaching the water table, its movement is predominantly horizontal but a downward component remains, which accounts for the tilt in V's vertical axis. Recharge originating between two dates occupies an oblique prism bracketed by elevations z_1 and z_2.

We could estimate the water year's solute load as long as virtually all the water-year interval was within the monitored zone. Since groundwater was flowing horizontally past MLPs between sampling events, later events encountered groundwater that had originated farther upgradient than earlier events. Thus, water-year profiles sampled on different dates represented slightly different locations in the field, providing some representation of within-field variability.

Nitrogen Budget

We used the budget method of Meisinger and Randall (1991) to calculate NO_3–N loading according to:

$$N_{pl} = N_{input} - N_{output} - \Delta N_{st} \qquad [3]$$

where N_{pl} is long-term potentially leachable total nitrogen, N_{input} and N_{output} are N entering and leaving the field between the top of the crop canopy and the bottom of the root zone, and ΔN_{st} is the change in N storage. We used N_{pl} as our budget-derived estimate of NO_3–N loading to groundwater during the period between successive plantings. This is justified because NO_3 and other nonadsorbing solutes have been found to leach rapidly (within months) to groundwater in study-area soils, especially where the vadose zone is thin (Harkin et al., 1986; Kung, 1990a).

N_{input} components considered were fertilizer, atmospheric deposition, and crop seed. Fertilizer amounts were obtained from growers' records. Atmospheric N deposition was estimated at 20 kg ha^{-1} yr^{-1}, based on a review of work by Hoeft et al. (1972) and others. Corn seed would supply a negligible 0.3 kg N ha^{-1}, and potato "seed" 9 kg ha^{-1} (Meisinger and Randall, 1991).

The largest N_{output} component was crop harvest, amounting to 108 kg ha^{-1} for 1992 sweet corn and 110 kg ha^{-1} for 1993 potato. We calculated harvested N from crop yield (see Study Field and Management Practices section, above) and N concentration. Meisinger and Randall (1991) tabulated N concentrations (dry matter) and moisture content for numerous crops, giving a "general range" and a "common value" near the middle of the range. For sweet corn, we adopted a N concentration of 4.3 g kg^{-1}, which is the common value converted to the concentration in fresh ears with husks (the general range is 3.8 to 4.9). Local N concentrations and moisture contents were available for potato (Saffigna et al., 1977; Bundy et al., 1997; Wilner et al., 1997), with a fresh-weight N concentration range of 2.0 to 2.8 g kg^{-1} and a mean of 2.4, which we adopted. Local potato N concentrations were considerably less than what Meisinger and Randall (1991) compiled from the literature. Outputs due to fertilizer volatilization and denitrification or erosion were judged to be negligible based on soil and aquifer properties (cf. Saffigna et al., 1977; Oberle and Bundy, 1987), and slight erosion hazard (Bartelme, 1977). Miscellaneous gaseous N losses from soil and crop are believed to be small based on Saffigna et al. (1977), and were estimated at 7 kg ha^{-1} for sweet corn and 9 kg ha^{-1} for potato (Meisinger and Randall, 1991).

ΔN_{st} has three potential components: inorganic N, crop residue, and soil organic matter. We assume that inorganic and crop-residue N are the same at the beginning and end of the budget year, and therefore ΔN_{st} from these is zero. This is consistent with measurements of the fate of N in crop residues in this irrigated vegetable cropping system (Andraski and Bundy, 1999). Therefore, only soil organic matter mineralization contributes to ΔN_{st}. Oberle and Keeney (1990) estimated N mineralization on Plainfield loamy sand to be 45 kg ha^{-1} yr^{-1} during the growing season. However, their method did not distinguish atmospheric N deposition from mineralization.

Accordingly, we decreased their estimate by the 20 kg ha^{-1} yr^{-1} atmospheric input, leaving $\Delta N_s = -25$ kg ha^{-1} yr^{-1}.

Chloride Budget

We computed a Cl budget similar to the N budget. Inputs considered were fertilizer (KCl and impurities in other fertilizers), atmospheric deposition, and irrigation water. Outputs were crop harvest and loading to groundwater. As with inorganic and residue N, we assumed that the change in Cl storage was zero. Chloride inputs from KCl fertilizer were 99 kg ha^{-1} in 1992 and 353 kg ha^{-1} in 1993. Input from impurities in other fertilizer materials was on the order of 1 kg ha^{-1} yr^{-1} or less (Hignett, 1985; J.P. Voth, personal communication, 1999). Various authors have reported atmospheric Cl inputs in the area from 1 to 7 kg ha^{-1} yr^{-1} (Cabrera-Rivera, 1989; MacDonald et al., 1992; Quideau and Bockheim, 1997). Chloride input from irrigation water was between 3 and 30 kg ha^{-1} yr^{-1}, probably toward the low end of that range. Crop harvest is the only significant Cl output other than leaching. Sweet corn Cl concentrations (W.L. Bland, personal communication, 2000) indicate crop-harvest output of 8 to 10 kg ha^{-1}. Potato Cl concentrations inferred from Saffigna et al. (1977) imply harvest outputs of 22 to 37 kg ha^{-1}.

RESULTS

Water-Year Loading

Chloride breakthrough for the 1992 water year began in July of that year. The 1993 breakthrough began by April in one MLP. Wet conditions prevented sampling in May and June, and breakthrough had begun by July in all other MLPs. Water-year intervals typically remained in the monitored zone about 200 d, during which loading could be estimated from an average of five sampling events.

A separate estimate of NO_3–N and Cl loading was derived from each sampling event containing a complete or nearly complete water-year interval at each MLP (Table 1). Loading observations for each water year were averaged across sampling events in individual MLPs, and then a grand mean was calculated for the six MLPs. This revealed average loadings for the 1992 water year, when sweet corn was grown, of 165 kg ha^{-1} of NO_3–N and 111 kg ha^{-1} of Cl. In 1993, with potato, water-year loadings were 228 kg ha^{-1} of NO_3–N and 366 kg ha^{-1} of Cl. The maximum difference among MLPs in average NO_3–N loading in a given water year was 70 kg ha^{-1} in 1992 and 74 kg ha^{-1} in 1993 (Table 1). Some variation is explained by unequal N applications and field-edge effects. The north field half received 60 kg ha^{-1} less fertilizer N than the south half in 1993 because of poor trafficability. Loadings that year averaged 48 kg ha^{-1} less NO_3–N in north-half MLPs (#1 and #2) than in the south half. We cannot be certain how much of the loading reduction resulted from smaller fertilizer applications, because of a confounding effect wherein mixing of upgradient groundwater influenced the deeper ports of the two northern (more upgradient) MLPs (Stites and Kraft, 2000), despite our efforts to place all MLPs more than one year's groundwater travel distance from field edges. However, the difference in NO_3–N loading between the north and south halves averaged only 7 kg ha^{-1} in 1992 when fertilizer was applied uniformly, supporting the possibility that the half receiving less N fertilizer in 1993 loaded less NO_3 to groundwater. The range of Cl loading estimates among MLPs spanned 69 kg ha^{-1} in 1992 and 126 kg ha^{-1} in 1993.

The temporal range in NO_3–N loading among sampling events at the same MLP was about the same magnitude as the range among different MLPs, averaging 60 to 75 kg ha^{-1} each year. This temporal variation was at least partly due to spatial variability, because samples collected on different dates originated at different positions upgradient of MLPs. Hence, temporal variation in estimated loading reflected nonuniformity in fertilizer application or processes that redistribute Cl or N. The range in Cl loading estimated on different dates varied

Table 1. Nitrate N and Cl water-year loading for 1992 and 1993.

MLP†	Dates‡	Observations§	Nitrate N: Mean Loading	Nitrate N: Range	Chloride: Mean Loading	Chloride: Range
			kg ha^{-1}			
			1992			
1	260–560	7	184	128–243	87	77–91
2	209–463	8	137	81–190	88	75–100
3	552–596	2	144	140–148	129	122–135
4	450–560	2	207	185–230	156	121–191
5	260–463	6	160	134–183	90	62–127
6	582–721	4	157	144–175	115	94–142
Mean			165		111	
Std. error			11		11	
			1993			
1	216–482	7	205	143–253	338	276–449
2	252–482	6	187	169–208	363	185–626
3	287–482	5	242	209–303	399	360–452
4	287–482	5	261	227–308	448	325–630
5	287–482	5	224	203–252	328	232–431
6	477–477	1	250	–	322	–
Mean			228		366	
Std. error			12		20	

† Multilevel piezometer number.
‡ Days after the beginning of the calendar year.
§ Number of sampling events when the monitored zone contained a complete enough water-year profile to permit calculation of the annual loading.

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Table 2. Summary of 1992 and 1993 N budgets, including budget-derived N loading. Water-year N loading is provided for comparison. Best values for N outputs are based on the most likely harvested crop N concentration; the range given is based on literature values. Separate estimates are shown for the field halves in 1993, when halves were managed differently.

			Harvest + gaseous losses		N loading		
					Budget-based		Water-year
Field-half	Crop	Available N†	Best	Range	Best	Range	Mean
		kg ha^{-1}					
			1992				
Both	sweet corn	295	116	104–131	179	164–191	165
			1993				
North	potato	351	118	100–136	233	215–251	196
South	potato	411	118	100–136	293	274–311	233
Mean	potato	381	118	100–136	263	244–281	228

† Fertilizer, precipitation, and mineralized N.

from 13 kg ha^{-1} at MLP 3 to 70 kg ha^{-1} at MLP 4 in 1992 and from 92 kg ha^{-1} at MLP 3 to 441 kg ha^{-1} at MLP 2 in 1993. The larger ranges of 1993 corresponded with the 3.5-fold larger 1993 Cl application.

Nitrate and Chloride Loading from Budget

Budget-derived NO_3–N loading was 179 kg ha^{-1} for the 1992 sweet corn crop, and 233 (north field-half) or 293 (south) kg ha^{-1} for the 1993 potato crop (Table 2). These values are 8% more than the water-year NO_3–N loading in 1992 and 13% greater in 1993 (north and south field halves averaged for 1993). Given the uncertainty in N-budget parameters—especially crop N concentration—and the variability among MLPs, the agreement between methods is remarkable.

Budget-derived Cl loading ranges based on the extremes of the input and output ranges were 95 to 128 kg ha^{-1} for 1992 sweet corn and 320 to 371 kg ha^{-1} for 1993 potato. Budget-derived Cl loading was 86 to 115% of water-year loading in 1992, and 87 to 101% in 1993. The water-year and budget loadings appear to be in good agreement.

DISCUSSION

The water-year method was successful in this setting for measuring NO_3 and Cl loading to groundwater. Attributes of the physical setting that promote leaching of solutes to groundwater during the year of application (humid climate, coarse soils, and shallow groundwater) contributed to the method's success. The water-year method might not work as well where the vadose zone is thicker, because a thicker vadose zone could give rise to greater solute dispersion or large-scale preferential flow paths (e.g., funnel flow; Kung, 1990b), causing the Cl pulse in groundwater to be less distinct. The method could be impractical where the vadose zone is tens of meters thick or where finer-textured sediments limit the rate of solute loading to groundwater.

An important question is how closely the measured N loading during a water year corresponds with the N leached by that year's crop. They are not necessarily identical, because a crop may contribute some leachable N that does not appear during the same water year. In particular, this kind of discrepancy could develop if mineralization of crop-residue N is delayed and the N content of the residues differs substantially between successive years. In the conditions of this study, the potato vines and sweet corn stalks decompose rapidly and their N is largely mineralized before winter. Almost none of the crop-residue N is available to the following year's crop; even a fall cover crop was found to recover only about 2 kg N ha^{-1} from the previous crop's residue and leftover fertilizer combined (Bundy et al., 1997). Moreover, the difference in average residue nitrogen content for these two crops is rather small compared with the entire nitrogen budget: sweet corn residue averages about 90 kg N ha^{-1} and potato residue about 60 kg N ha^{-1} (Bundy et al., 1997; Andraski and Bundy, 1999). Thus it is reasonable that the water-year loading estimate closely approximated leachable N estimated from the budget. The potential for a discrepancy between crop N leaching and leaching during a fixed time period applies to any N loading measurement, except perhaps isotope tracer methods.

The water-year method for measuring solute loading to groundwater has certain advantages over methods that rely on vadose zone monitoring. Loading is measured directly in groundwater rather than inferred from extracted soil or from soil water samples. Thus, bypassing of samplers by preferential flow and difficulty of estimating water flux become nonissues. Problems due to soil disturbance are also minimized because horizontal groundwater flow transports water to the MLPs from outside the disturbed zone. Spatial variability affects solute loading estimates by the water-year method, but at larger scales than vadose zone methods. Loading estimates derived from vadose zone methods encounter variability at the scale of soil samples or porewater sampling devices. In the saturated zone, recharge from a larger volume is integrated and sampled by MLPs. In this study, six MLPs appear to sufficiently represent the study field, as the standard errors are only 6 to 10% of the mean (Table 1).

The agreement between NO_3 loading by the budget and water-year methods bolsters confidence in both as tools for measuring and predicting NO_3 loading. The budget-derived NO_3 loadings were slightly (2–8%) greater. Agreement between methods was also good for Cl. Budget-derived loadings were sensitive to the concentration of N in the harvested crop which, for potato, has a wide range in the literature. If the average potato-tuber N concentration reported by Meisinger and Randall (1991; 4.0 g kg^{-1}) had been used instead of a locally based concentration (2.4 g kg^{-1}), the budget loading estimate would have been 189 kg ha^{-1} rather than 263 kg ha^{-1}. Local data are important if accurate results are to be obtained from the budget method.

Nitrate loading to groundwater constituted nearly the same proportion of N input both years, even though much more fertilizer N was applied in 1993 than in 1992. In 1992, water-year NO_3–N loading was 66% of fertilizer N or 56% of available N (defined as the sum of fertilizer, precipitation, and mineralized N), and in 1993 it was 70% of fertilizer N or 60% of available N (Table 2). Nitrate loading from the 1992 sweet corn crop was less than from 1993 potato, mainly because potato was fertilized more heavily.

Comparison of this study's sweet corn loading with others is difficult because the literature is scant. Far more loading literature is available for field corn, but it may not be applicable. A sweet corn study by Tucker and Hauck (1978) reported 77 kg ha^{-1} N harvested with fertilizer applications of 116 to 222 kg ha^{-1}, implying N losses of 39 to 145 kg ha^{-1} or 34 to 65% of fertilizer N. Our loading rate, 165 kg ha^{-1}, is 114% of the maximum inferred from Tucker and Hauck, but our fertilizer N input was 113% of their maximum. Our N loading as a proportion of fertilizer N, 66%, matches the upper end of their range.

Most studies of N loading from potato have reported smaller rates than we found. Milburn et al. (1990) and Madramootoo et al. (1995) report NO_3–N leaching of 5 to 70 kg ha^{-1}, or 5 to 38% of fertilizer N, in drain-tile effluent with fertilizer rates of 110 to 200 kg N ha^{-1}. These estimates were probably low, because winter and early spring leaching, which can be important (Hill, 1986), was not captured. Also, denitrification (likely where tile drainage is used) may have decreased NO_3 available for leaching in those studies. In a soil-coring study, Hill (1986) reports a 3-yr average N loading of 134 kg ha^{-1}, which was 70% of the inputs from fertilizer, precipitation, and irrigation, similar to the proportion we found. Perillo et al. (1993) used porous-cup samplers at a Minnesota sand-plain site, and found N leaching losses were 18 to 69 kg ha^{-1}, 25 to 30% of the fertilizer N input of 90 to 269 kg ha^{-1}. Their samplers were probably not operable in winter, so these leaching amounts are likely to be underestimates. The most comparable study to ours was that of Saffigna et al. (1977) because it was conducted in the same area and had the most complete measurement of N fluxes. They measured 120 to 215 kg ha^{-1} NO_3–N leaching in monolith lysimeters, an average of 58% of fertilizer + irrigation N, or 47% of available N. These leaching amounts and proportions are somewhat less than ours, probably because our study field received larger fertilizer inputs.

The agreement between measured (water-year) and budget-derived NO_3–N loading suggests that the budget method can be used for estimating loading for Wisconsin central sand plain vegetable production systems. For sweet corn, the typical fertilizer N input is about 200 kg ha^{-1} (W. Ebert, personal communication, 1997), and the University Extension recommended N input is 168 kg ha^{-1} (Binning et al., 2000). Based on a long-term average experimental-farm yield of 20 Mg ha^{-1} (L.G. Bundy, personal communication, 1999), the budget-derived NO_3–N loading is 151 kg ha^{-1} for typical input and 119 kg ha^{-1} for recommended input. For potato, the typical fertilizer N input (equal to the University Extension recommendation) is 258 kg ha^{-1} (J. Exo, personal communication, 1993). Based on a potato harvest of 41.5 Mg ha^{-1} (1992–1996 average; National Agricultural Statistics Service, 1998), the loading is 203 kg ha^{-1} for the typical (or recommended) input.

A modest groundwater quality goal might be that basin-averaged NO_3–N does not exceed the 10 mg L^{-1} MCL. One way to attain this goal is by diluting the NO_3 in groundwater recharge from potato and sweet corn fields with nitrate-free recharge from other land uses. (Groundwater NO_3 is generally conserved in this setting [Kraft et al., 1999], so natural denitrification in the aquifer is not a consideration.) Given the annual groundwater recharge rate of 240 mm (Holt, 1965), NO_3–N loading amounts greater than 24 kg ha^{-1} yr^{-1} will cause average groundwater quality to exceed the MCL. The loading amounts of 119 to 203 kg ha^{-1} expected with recommended fertilizer N inputs to sweet corn and potato are five to eight times that amount. Hence, each hectare of irrigated sweet corn and potato would require 5 to 8 ha of land uses with zero NO_3 loading to meet even this modest groundwater quality goal. Previously, Stites and Kraft (2000) estimated that the NO_3 goal could be attained if each hectare of sweet corn and potato were balanced by only one hectare of zero-NO_3 loading land uses, based only on NO_3 concentrations observed beneath irrigated fields. That ratio was acknowledged to be an overly optimistic projection, due to some simplifying assumptions. The revised estimate highlights the value of determining NO_3 loading rather than using simple concentration information when making water quality assessments and projections.

CONCLUSIONS

Irrigated sweet corn and potato loaded substantial NO_3 to groundwater. Measured NO_3–N loading was 165 kg ha^{-1} for 1992 sweet corn and 228 kg ha^{-1} for 1993 potato, equivalent to 66 to 70% of fertilizer N and 56 to 60% of available N (fertilizer + precipitation + mineralized N). These amounts are greater than loading under both typical and university-recommended fertilizer N applications. Budgets indicated that NO_3 loading under sweet corn is 151 kg ha^{-1} with typical management and 119 kg ha^{-1} with the University Extension recommended fertilizer rate. Under potato, calculated NO_3–N loading with the typical (equal to recommended) fertilizer input is 203 kg ha^{-1}. Use of recommended fertilizer rates reduces NO_3 loading little or none compared with the typical input.

In the study area, NO_3–N loading greater than 24 kg ha^{-1} yr^{-1} causes average groundwater quality from fields to exceed the 10-mg L^{-1} MCL, but the loading expected with typical or university-recommended fertilizer-N input is five to seven times that amount. Dilution is the only significant mechanism decreasing groundwater NO_3 concentrations in this setting. To meet the groundwater-quality goal through dilution, each hectare of irrigated sweet corn and potato would have to be offset by more than 5 to 8 ha of land uses that contribute no NO_3 to groundwater recharge.

The water-year method worked well for measuring NO_3 loading under a field in the Wisconsin central sand plain. Agreement between water-year and budget methods for both NO_3 and Cl supported the validity of both methods, and our results also agreed with limited information available from other studies (Saffigna et al., 1977; Tucker and Hauck, 1978). The water-year method provided a direct measurement of loading below the field, and appears less affected by spatial variability and preferential flow than some other methods. No other method appears as well adapted for measuring field-scale loading in this setting. A limitation to the method may be that dispersion in a thicker vadose zone could smear out concentration gradients required to distinguish annual pulses. Future studies could establish under what conditions the method remains applicable.

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Controlling Nitrate Leaching in Irrigated Agriculture

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ABSTRACT

The impact of improved irrigation and nutrient practices on ground water quality was assessed at the Nebraska Management System Evaluation Area using ground water quality data collected from 16 depths at 31 strategically located multilevel samplers three times annually from 1991 to 1996. The site was sectioned into four 13.4-ha management fields: (i) a conventional furrow-irrigated corn (*Zea mays* L.) field; (ii) a surge-irrigated corn field, which received 60% less water and 31% less N fertilizer than the conventional field; (iii) a center pivot-irrigated corn field, which received 66% less water and 37% less N fertilizer than the conventional field; and (iv) a center pivot-irrigated alfalfa (*Medicago sativa* L.) field. Dating ($^3H/^3He$) indicated that the uppermost ground water was <1 to 2 yr old and that the aquifer water was stratified with the deepest water ~20 yr old. Recharge during the wet growing season in 1993 reduced the average NO_3–N concentration in the top 3 m 20 mg L^{-1}, effectively diluting and replacing the NO_3-contaminated water. Nitrate concentrations in the shallow zone of the aquifer increased with depth to water. Beneath the conventional and surge-irrigated fields, shallow ground water concentrations returned to the initial 30 mg NO_3–N L^{-1} level by fall 1995; however, beneath the center pivot-irrigated corn field, concentrations remained at ~13 mg NO_3–N L^{-1} until fall 1996. A combination of sprinkler irrigation and N fertigation significantly reduced N leaching with only minor reductions (6%) in crop yield.

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Assessments of ground water NO_3 contamination report that many major areas of nonpoint-source contamination are located in the irrigated semiarid and arid regions of the western USA (Madison and Brunett, 1985; Anderson, 1989; Power and Schepers, 1989; Spalding and Exner, 1993). The thrust of these reports provides a clear association between nonpoint-source ground water NO_3 contamination and irrigated agriculture.

Irrigated agriculture has a major impact on the economy of several states both in the west and in the western Corn Belt. In Nebraska, 3.4 million ha of irrigated agriculture and related spin-off service industries add approximately $3 billion annually to the state's economy. Controlling leachates from irrigated crop land, especially from the 2.3 million ha of irrigated corn, requires fundamental changes in farm practices that not only lead to solutions but are acceptable to producers and regulators. Nebraska's dependence on ground water as the primary source of potable water is the major thrust for a sustained impetus to develop and implement more effective agricultural management strategies to reduce ground water NO_3 contamination.

Beginning in 1990, the USDA sponsored Management Systems Evaluation Area (MSEA) projects in five midwestern states in the corn and soybean [*Glycine max* (L.) Merr.] belt. The projects concentrated both on understanding the mechanisms involved in nonpoint-source contamination of surface and ground water by agrochemicals and on developing economically accept-

Abbreviations: MLSs, multilevel samplers; ET, evapotranspiration; MSEA, Management Systems Evaluation Area; NE-MSEA, Nebraska MSEA; CPNRD, Central Platte Natural Resources District; MCL, maximum contaminant level; DOC, dissolved organic carbon.